Spatial Conservation Prioritization in Data-Poor Areas of the World

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Abstract

Systematic conservation planning and spatial conservation prioritization are closely related fields of conservation science. They are concerned with the spatial allocation of conservation resources into actions such as expansion of reserve networks and allocation of habitat restoration, management or maintenance. Decision analytic techniques including optimization are heavily used, with the implication that sufficient data must be available to allow analyses that have relevance for on-the-ground planning. Most commonly, analyses are based on data about species distributions, costs and threats. However, in many parts of the world, including much of the tropics, comprehensive data about species distributions is not available. Are there feasible and robust methods for the allocation of conservation action when data is limited and species-based systematic conservation planning is not feasible? In this work I discuss spatial conservation prioritization and its application when data is limited - making a contrast to the case where ideal data is available.

Key words: Conservation Resource Allocation, Ecosystem, Robust Conservation Strategies, Site Selection, Systematic Conservation Planning, Threats.

Spatial Conservation Prioritization and Relatives

Systematic conservation planning (SCP) is a well-known framework in which biodiversity distribution is analysed, gaps are identified in the present reserve network, targets for additional representation of species are set, site selection optimization methods are applied, on-the-ground conservation action is planned together with stakeholders and conservation success is monitored; recent developments of this framework are described by Margules & Sarkar (2007), Sarkar & Illoldi-Rangel (2010) and Linke *et al.* (2011). SCP analyses are based on the so-called biodiversity features. The most common biodiversity feature is a species, but analyses can equally be based on habitat types (communities, ecosystems), ecosystem services, or even genes (Diniz-Filho *et al.* 2012), or combinations of several different types of features (see Loyola *et al.* 2011).

Spatial conservation prioritization is a related field of conservation science (Moilanen *et al.* 2009a) that has one poorly known yet relatively major difference with SCP. SCP is strongly focused on the identification of targets for biodiversity features. In SCP, the so-called reserve selection or site selection optimization techniques are applied to

satisfy these targets with minimum cost, thereby developing cost-efficient solutions (Margules & Sarkar 2007, Sarkar & Illoldi-Rangel 2010). Taking a different approach to spatial conservation, there exist methods for spatial priority ranking, which operate on the principle that generic relationships are defined for how conservation value should be aggregated across many features, space and time (e.g. Moilanen 2008; Moilanen *et al.* 2005, 2011a). These relationships can be converted into mathematical and algorithmic form without the specification of any hard targets for features.

Operationally and conceptually, this distinction is highly relevant, because recent evidence suggests that specification of hard targets to many low-level components of biodiversity (i.e. quantities such as representation levels, connectivity distances or population counts for species - total biodiversity value would be an aggregate of across tens of thousands of such low-level components) may easily lead to solutions that are inefficient in terms of return on investment (Moilanen & Arponen 2011; Di Minin & Moilanen 2012). This is so because the nested hierarchy of feature distributions cannot easily be accounted for in target setting. In concrete terms, disproportionate resources may end up being allocated to features that occur in generally feature-poor regions (Moilanen & Arponen 2011). Also variation in land cost is not usually accounted for in target setting, potentially further skewing resource use (Moilanen & Arponen 2011).

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There is another questionable feature in target setting: targets may falsely imply adequacy, meaning that socio-politically acceptable targets, which usually are much less than 1.0 (a representation target of, say, 0.3, implies protection of 30% of the occurrences of the feature), fail to guarantee long-term species persistence when it is in fact known that there already is an extinction debt for many ecosystems (Kuussaari *et al.* 2009). Habitat loss and degradation have led to a major global extinction debt, with the implication that what is presently left is not enough to guarantee persistence of all species, thereby implying habitat restoration and targets greater than 1.0 (Kuussaari *et al.* 2009). The global conservation area network is biased and inadequate (Gaston *et al.* 2008), and small additions to it might not stop species extinctions. The question arises: do targets really implement the fundamental principle of adequacy? A related fact is that targets can only be set for features for which distribution data is available. Issues such as these underlie the need for spatial conservation planning methods that are not based on the specification of many targets for biodiversity features.

Global and national conservation priorities are currently relevant with respect to the implementation of the resolutions of the tenth UNEP Convention on Biological Diversity, which was held October 2010 in Nagoya, Japan (Normile 2010; Harrop 2011). In this resolution it was stated that the coverage of terrestrial conservation areas should be raised from 10% to 17%. It was also stated that conservation efforts should be developed in a scientifically justified manner, thereby increasing demand for spatial conservation planning methods. However, what if these techniques have to be applied to regions where data is sparse, including much of the tropics and Biodiversity Hotspots such as the Brazilian Atlantic forest? Here, I focus on contrasting spatial conservation when data is abundant compared to when data is sparse. I also address the question of simple strategies that are likely to produce sensible outcomes and which can be applied even with a limited research background (Di Minin & Moilanen 2012) and under socio-political constraints, conditions present in many tropical countries (Diniz-Filho & Loyola 2010). If data is scarce, one may also ask if conservation efforts should be initiated immediately or should more data be collected first (Grantham *et al.* 2009). Investigating external costs and consequences of extractive land uses reveals that there is little time to waste and that the costs of a degrading environment are not fully accounted when short-term economic decisions are made (Sukhdev 2009).

Planning with Ideal Data

Table 1 summarizes data that can be used in spatial conservation planning: ideally, quite a bit of information should be available. Biodiversity is simplest protected where it occurs, and therefore data about distributions of biodiversity features (species, habitats, ecosystem services)

is a fundamental input into prioritization analyses (e.g. Naidoo *et al.* 2006; Sarkar *et al.* 2006; Nelson *et al.* 2009).

When aiming at cost-efficient solutions data about direct and indirect costs are needed as well (Pressey *et al.* 2007; Nelson *et al.* 2009). Threats influence landscape-wide retention of biodiversity and the persistence of biodiversity in the future (Pressey *et al.* 1994; Pressey & Logan 1998). It makes a great difference whether a threat can actually be stopped by conservation action. Some threats such as climate change are not influenced by local-scale conservation action, and others, such as forest clearing, can easily relocate if one land parcel is protected. Constraints, such as the budget or land availability influence the implementation of conservation.

Lastly, human preferences influence analysis objectives, analysis structure, weights given to features and implementation. A useful analysis can be done without information about all of the over thirty quantities and data components of Table 1, and publicly available decision support tools exist for both target-based systematic conservation planning (see Sarkar & Illoldi-Rangel 2010 for references) and for spatial priority ranking (see Moilanen *et al.* 2011a for references). But, what can be done if even species distribution data is not available?

Environmental Surrogacy and Planning with Scarce Data

An important factor in conservation planning is the availability of remote-sensing data about the environment. While observations of species occurrence can be sparse and taxonomically and geographically biased, high-resolution data about environmental conditions are becoming increasingly available (NOAA 2012). Also, software is available that can do large-scale high-resolution spatial prioritization on large GIS-derived grids (Arponen *et al.* 2012). Therefore, in this essay I shall emphasize a particular class of statistical modeling and surrogacy which allows statistically justified conservation planning with lower data demands than those associated with species-level analysis.

This class of analysis is macro-ecological analysis using models for species richness and turnover (Ferrier & Guisan 2006; Arponen *et al.* 2008; Guisan & Rahbek 2011; Mokany & Ferrier 2011). Arponen *et al.* (2008) identified the components needed for successful ecological community-level decision analysis, including statistically justified simultaneous handling of species richness and pair-wise similarity of ecological communities, modeled as functions of environmental conditions. This approach works for data-poor systems, because the number of statistical models needed is very low compared to species-level analysis. Instead of needing models for perhaps tens of thousands of species, one only needs two models (richness and turnover) per species group (guild or higher taxon) – a several order of magnitude difference in model count. Additionally, community-level models fitted elsewhere could be used as priors in the

Table 1. Data that can be profitably used in spatial conservation planning. Even under the best scenario, only part of this data will be available. Simplest analyses only include information about species distributions. Note: this table should not be taken as exhaustive.

Major category	Type	Component	Usage
Biodiversity distribution data	Species	Present distribution	Most common quantity of spatial planning.
		Future predicted distribution	Predicted assuming climate change; used for identifying future core areas of biodiversity.
		Historical distribution	Sets baseline; degree of decline to present time may influence weight given for feature.
		Connectivity responses	Relevant for understanding the importance of habitat pattern for the species.
		Persistence	Is required for favourable conservation outcome. Based on habitat area, quality and connectivity, but difficult to estimate.
		Genetic diversity	May influence health of populations. Large-scale data for many species hard to come by.
		Vulnerability to threats	What kinds of human-caused pressures is the feature sensitive to?
		Environmental niche	Identified via species distribution modelling; basis of statistical distribution modelling.
	Habitat types, ecosystems	Present, historical, & future predicted distribution	As for species, but feature is a habitat type or ecosystem with specific characteristics. A commonly used feature type.
		Vegetation condition; vulnerability	Habitat condition is reduced by human impacts; highly important quantity but not necessarily easy to estimate.
		Similarity to other habitat types	Two nominally different habitat types (say spruce forest and pine forest) may share a significant fraction of species, which should be accounted for in connectivity calculation.
	Ecosystem services	Distributions	Where are they generated?
		Underlying ecosystem processes	What ecosystem processes produce the ecosystem service?
		Value to society	Value given to specific ESS.
Costs & benefits	Direct costs	Land acquisition	Cost of buying land for conservation.
		Maintenance	Cost of maintaining conservation areas in good condition.
	Indirect, opportunity costs	Costs to different stakeholders	Conservation may cause different opportunity costs to different stakeholders.
	Benefits	Ecosystem services	Value of benefits provided by ESS that are maintained by conservation action.
		Direct benefits and income from nature	Different material and immaterial benefits to local communities from habitats and species maintained by conservation.
Threats	Type; effects of conservation action	Stoppable	The threat can be removed by conservation.
		Unstoppable	Conservation action will not remove the negative effects of the threat. Like, climate change is not stopped by local action.
		Transferrable	The threat (e.g., forest clearing) can be stopped in one area but it will relocate nearby.
	Effect on features		Sensitivity of features to threats.
Feasibility, constraints and opportunities	Intensity distribution		Spatial distribution of the intensity of the threat.
	Budget		Amount of resources available for conservation. Could also be specified as yearly allocations.
	Land availability		Availability of land parcels for conservation; conservation opportunities.
	Governance		Influences likelihood of successful implementation of conservation
	Planning units	Property and administrative borders	Division of landscape into units that should be uniformly treated in planning. Preferences may differ between administrative areas.
Human preferences		Hydrological catchment structure	Threats may be correlated within hydrological catchments making them natural planning units at least for riverine analyses.
	Preferences of people and institutions		will influence analysis structure and the relative weights or targets assigned to different features.
Environmental variables			While conservation decisions rarely are directly based on environmental factors, data about environmental variables underlies statistical distribution modelling.

Bayesian sense, further facilitating well-informed analysis. For example, the influence of temperature and rainfall on species richness could be similar in Asia and South America. Note that communities, ecosystems, environment types and habitat types are all entities that can analogously be analyzed using this approach.

Surrogacy is a major theme in conservation biology (Cabeza *et al.* 2008; Grantham *et al.* 2010). It is essentially about the investigation of the co-occurrence of biodiversity features, aiming to use distributions of easily observable features as surrogates for those features for which distribution information is not available and cannot be modeled. Most frequently, the conclusion has been that species-level surrogates work only partially (Rodrigues & Brooks 2007; Kremen *et al.* 2008), with the implication that a lot of species data across many taxa is needed if spatial conservation planning is to cover the needs of all species. For example, one could not assume that amphibians should be a good surrogate for occurrence of birds. On the other hand, there are individual studies that show helpful performance for some surrogates (or indicators): for example Loyola *et al.* (2007) found that endemic vertebrates were a valuable surrogate for conservation prioritization in Brazil. Community-level conservation prioritization can be interpreted as ecologically informed use of environmental surrogates - a completely different class of surrogacy than surrogacy between species.

Publicly available methods exist for community/ecosystem/habitat/environment level spatial prioritization (Leathwick *et al.* 2010; Moilanen *et al.* 2011b). The general idea is to protect a balanced set of environmentally diverse areas that are relatively species-rich and in relatively good condition (low human impacts). In this process, communities are derived from a model of species turnover as a function of changes in environmental conditions. Each community receives a species richness value, which can be observed or modeled. Spatial prioritization then proceeds, accounting for the species richness of each community, the ecological (dis-)similarity of pairs of communities and landscape/vegetation condition. Human impacts including projected habitat loss can be linked via changes in the vegetation condition layer. It is even possible to account for climate change in somewhat complicated variants of these analyses (Sommer *et al.* 2010; Mokany & Ferrier 2011). Proximity to existing reserves or avoidance of threat or invasive species can be modeled via a connectivity interaction (Rayfield *et al.* 2009).

Figure 1 compares the process of species and community level conservation prioritization. Both approaches combine landscape-wide information about environmental variables with species observations. Species-level modeling is done separately for each species, with the limitation that models for rare species are hard to estimate due to a low number of observations leading to low statistical power. As a major difference, rare species become naturally integrated in community-level modeling, which is based on information about species lists collected at sample locations (Ferrier &

Guisan 2006; Guisan & Rahbek 2011; Mokany & Ferrier 2011). For example, rare species are naturally counted in the species richness of a location, which is the response variable of a richness model. In community-level analysis human impacts could be included via aggregate variables such as vegetation condition, but in species-level analysis human impacts would need to be accounted for separately for each species. Overall, there are many benefits for the community/ecosystem pathway of spatial prioritization. But, what if data for spatial planning simply does not exist, if time is insufficient for any kind of quantitative analysis, or if human resources are unavailable?

Robust Conservation Strategies

An ideal conservation strategy produces a highly favorable and cost-effective outcome with high reliability (Moilanen *et al.* 2006). A poor strategy is one that produces a poor outcome with certainty. Robustness analysis is needed when estimated benefits are high, but uncertain (Moilanen *et al.* 2006). If known benefits are low but uncertainty is favorably biased, there is scope for positive surprises - this could be the case of poorly surveyed areas. Dozens of very different conservation strategies have been proposed in scientific literature, including such as “increase connectivity”, “translocate species”, “practice adaptive management”, “study responses of species to climate change”, and so on (Heller & Zavaleta 2009); it is highly relevant to ask which ones are robust and reliable and under what circumstances (Hodgson *et al.* 2009). Assuming limited information, what kinds of considerations are helpful in guiding us towards robustly sensible use of conservation

Figure 1. Pathways for species-centered and community/ecosystem-centered distribution modeling and spatial prioritization.

resources? It is important to be able to grasp conservation opportunities and act quickly (Cowling *et al.* 2010), and simple strategies allow rapid action.

An example of a strategy that is not automatically robust can be found in naive implementation of biodiversity offsetting (e.g. Stokstad 2008; Walker *et al.* 2009). Assume, for example, that a development project causes immediate and complete loss of, say, natural-state rain forest. Assume that this loss is to be compensated via offsetting by habitat restoration. At this stage, the examination of uncertainties reveals that habitat restoration is by no means guaranteed to succeed (Suding 2011), that clearing of natural or semi-natural vegetation might be best avoided (Gibbons & Lindenmayer 2007), and that offset measures have frequently failed to successfully compensate for lost habitats (Moilanen *et al.* 2009b; Walker *et al.* 2009). So, what kinds of strategies could be robust? While an exhaustive investigation of this topic is worth an entire research program, some illustrative ideas can be provided here.

One question that frequently arises is the relative role of connectivity compared to habitat area and quality. Should one invest in increased area of reserves or should we more specifically target improved connectivity? Hodgson *et al.* (2009) provide a perspective to this question in the context of multi-species conservation. Imagine a landscape of variable environmental conditions and human impacts. Different pieces of the landscape will be suitable for the reproduction or dispersal of individual species. Habitat quality is linked to low human impacts and disturbance. High quality areas can be used by many species for reproduction and by even more species for dispersal. But what about area, quality and connectivity? The relative role of connectivity is an important question because improving connectivity is the strategy most commonly proposed for improving conservation under climate change (Heller & Zavaleta 2009).

It is a combination of area and quality that determines the regional carrying capacity for a species, and increasing area or quality will increase carrying capacity (Hodgson *et al.* 2009). On the other hand, connectivity, which is closely related to spatial aggregation, determines what fraction of the carrying capacity can be utilized. Connectivity is a function of area and quality - there is no connectivity without area. Also, there are very many different kinds of species-specific connectivity (Kindlmann & Burel 2008), making connectivity a comparatively uncertain decision criterion (Hodgson *et al.* 2009). Consequently, area and quality are primary considerations and connectivity only follows after. Empirical evidence supports the primary role of habitat quality for determining species occurrences (Mortelliti *et al.* 2010).

As a corollary of the importance of habitat quality: avoid all native clearing when possible, because habitat restoration cannot reliably regenerate pristine-state habitats, communities or ecosystem types (Gibbons & Lindenmayer 2007; Suding 2011). This conclusion is most relevant for slow-developing

late-successional habitats, such as old-growth forests or coral reefs. Natural or semi-natural habitats in global Biodiversity Hotspots are of primary conservation concern (Brooks *et al.* 2006; Ceballos & Ehrlich 2006; Lamoreux *et al.* 2006), and any clearing of them is likely to cause irreparable biodiversity losses, implying that investment into avoided deforestation (REDD) may be a valuable strategy (Harvey *et al.* 2009). In this context, remember both sides of the conservation coin. The obvious side is expansion of reserve networks to include larger areas of high-quality habitats. The other side is impact avoidance: when designing land-use zoning, try to guide environmentally harmful activity to the ecologically lowest-priority parts of the landscape (Moilanen *et al.* 2011a).

What about habitat heterogeneity? It has frequently been observed that heterogeneous habitats have higher species richness than homogeneous ones (Fahrig *et al.* 2011). Also, heterogeneous habitats provide relatively higher population stability than less heterogeneous landscapes (Oliver *et al.* 2010). Habitat heterogeneity could also be seen as one factor that may buffer against climate change, for example, reducing the need of species to move due to climate change (Hodgson *et al.* 2009; Mokany & Ferrier 2011). There is a perspective to this question as well: heterogeneity is desirable when the question is of a naturally heterogeneous habitat mosaic, such as a mosaic of forests and wetlands. Heterogeneity is less valuable when it has been caused by human action in a previously homogeneous habitat: constructing roads and building and clearing agricultural fields will turn a previously continuous forest into a more heterogeneous area - but with little true benefits for biodiversity. One could summarize that conservation is ultimately about avoiding or reversing human impacts on nature. Thus, one can ask: how can future impacts be reduced by present-day conservation action? How can action reverse past negative human impacts? How does biodiversity support us, and how can one promote multi-use landscapes that provide nature-derived income to many people while valuing biodiversity (Sukhdev 2009).

Conclusions

The fact is that data about biodiversity distribution will always be incomplete - except if in the future a major breakthrough is made in automated survey methods. Likewise, our understanding about the interactions between features will remain uncertain, human preferences about conservation will vary, and the effects of human action on habitat loss and climate change are subject to the uncertainties of global politics. Nevertheless, land use decisions that impact conservation must be made on a routine basis all around the world. Individuals, institutions and authorities responsible for conservation must be enabled to act. In the short term it may appear as if the society might not afford conservation due to the opportunity costs implied by it. But, accounting for the negative externalities of intensive extractive land uses and thinking in the long term, is why

we cannot afford not to conserve our environment (e.g. Sukhdev 2009; Harrop 2011).

This paper has been written as a high-level overview of considerations relevant for spatial conservation prioritization in data-poor areas of the world. It complements a previous perspective about systematic conservation planning (Sarkar & Illoldi-Rangel 2010), with the objective of giving the reader further ideas and references about how to approach spatial conservation. Observations worth remembering include that there are other, less data-hungry, approaches to conservation than the species-based one. There are spatial prioritization methods that do not require the specification of many low-level targets. When in doubt, look for robust strategies with high likelihood of securing some gains and low likelihood of outright failure.

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